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ORIGINAL PAPER

Temporal trends in the protective capacity of burnt beech forests (*Fagus sylvatica* L.) against rockfall

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Abstract Beech (*Fagus sylvatica* L.) forests covering relief-rich terrain often provide direct protection from rockfall for humans and their property. However, the efficacy in protecting against such hazards may abruptly and substantially change after disturbances such as fires, windthrows, avalanches and insect outbreaks. To date, there is little known about the mid-term evolution of the protective capacity in fire-injured beech stands. We selected 34 beech stands in the Southern European Alps that had burnt in different intensity fires over the last 40 years. We inventoried all living and dead trees in each stand and subsequently applied the rockfall model Rockfor.net to assess the protective capacity of fire-injured forests against falling rocks with volumes of 0.05, 0.2, and 1 m^3. We tested forested slopes with mean gradients of 27°, 30°, and 35° and lengths of 75 and 150 m. Burnt beech forests hit by low-severity fires have nearly the same protective capacity as unburnt forests, because only thin fire-injured trees die while intermediate-sized and large-diameter trees mostly survive. However, the protective capacity of moderate- to high-severity burns is significantly reduced, especially between 10 and 30 years after the fire. In those cases, silvicultural or technical measures may be necessary. Besides the installation of rockfall nets or dams, small-scale felling of dying trees and the placement of stems at an oblique angle to the slope can mitigate the reduction in protection provided by the forest.

Keywords Forest fires · Stand structure · Burn severity · Rockfor.net

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Introduction

In mountain regions, forests often provide direct protection from natural hazards for humans and their property (Dorren et al. 2005a; Brang et al. 2006). In comparison with man-built structures, the protective effect of forests is naturally renewing and relatively cost-efficient (Olschewski et al. 2012). In case of rockfall events, standing and fallen trees act as barriers against falling rocks (Motta and Haudemand 2000), and understory vegetation increases the surface unevenness, contributing to the forest stand's overall capacity to dissipate energy (Dorren et al. 2004b; Brauner et al. 2005). Whether the protection provided by a particular forest stand is effective or not is mainly determined by: (1) terrain characteristics and the total length of the forested part of a slope between the rockfall release area and the area to be protected, (2) the size and kinetic energy of the falling rock, and (3) the dendrometric characteristics of the trees within the forested area that reduce or adsorb the impact energy of falling rocks (Dorren et al. 2015).

Since forests are dynamic ecosystems, their protective capacity changes constantly. In particular, natural disturbances such as forest fires, windthrows, pests and insect outbreaks, and snow avalanches have the potential to abruptly and substantially reduce the protective capacity of a stand (Vacchiano et al. 2016). The influence of these factors on a stand's protective capacity highly depends on (1) the intensity and scale of the disturbance, (2) the resistance and resilience of the concerned stand, and (3) the post-disturbance management (Bebi et al. 2015). For instance, insect outbreaks or low- intensity windthrows can cause dispersed tree damage, which increases light and nutrient availability, to favor pre-regeneration (Kupferschmid Albisetti 2003; Collet et al. 2008; Kramer et al. 2014). In case of an immediate and comprehensive loss of living trees after the disturbance, remnant dead wood may significantly decrease terrain patency and may thus at least partly compensate for this deficit. However, slow succession rates after a disturbance event and relatively fast decay of dead wood may result in a time window of temporarily reduced protection against natural hazards (Bebi et al. 2015).

Fire affects both the pre-fire regeneration and the dead wood structure (Wohlgemuth et al. 2010), which may additionally reduce the protective capacity of burnt forests with respect to windthrow areas. Unfortunately, to date little is known about fire resistance and post-fire resilience of different forest types with potentially important protective functions. This is particularly true for European beech (*Fagus sylvatica* L.) forests, which hold a share of 16 % (Switzerland) and 26 % (Piedmont, Italy) of the overall forests that protect against rockfall in the southwestern European Alps (Brändli and Huber 2015; Istituto per le Piante da Legno e l'Ambiente 2012).

Healthy and well-structured beech stands are able to dissipate the kinetic energy from falling rocks (Perzl 2009; Schmidt 2005). Recent studies have demonstrated, however, that fire-injured beeches in moderate- and high-severity burns generally collapse within the first 20 years post-fire (Maringer et al. 2016), giving rise to simultaneous and enhanced seed germination and seedling emergence due to the increasing canopy opening and the removal of thick litter layers (Ascoli et al. 2015; Maringer et al. in press). Both processes highly depend on fire severity (i.e., the immediate effect of fire; cf. Morgan et al. 2014). In case of very severe fires, most beeches die within the first few post-fire seasons, causing a lack in seed production. Additionally, fast-growing early post-fire colonizers like shrubs, grasses and ferns tend to build dense layers inhibiting the growth of new seedlings (Maringer et al. in press). In contrast, after low-severity fires only a few individual (usually small) trees are critically injured with marginal consequences for the stand dynamics. Fires of intermediate severity cause a progressive dieback of the stand depending on the average proportion of the trunk injured and the proliferation of decay-causing fungi (Conedera et al. 2007, 2010; Maringer et al. 2016). Here the probability of successful seed germination and seedling emergence is highest, especially when a mast year immediately follows the fire event (Ascoli et al. 2015). All the described post-fire processes in beech stands show that there might be a temporary deficit in the forest protective capacity; particularly in moderate- and high- severity burns. It is thus crucial for forest managers to know about the influence of post-disturbance processes in order to prevent the associated risks.

In this paper we investigate the question of a possible reduction in the protective capacity of beech forests against rockfall as a consequence of mixed-severity forest fires. The main aim of this research is to support foresters in post-fire management decision-making (e.g., salvage logging, reforestation) in order to minimize the loss of the forest's protective function. To this purpose we assessed forest stand characteristics in beech forests burnt between 1970 and 2012 and we modeled their capacity to protect against rockfall in average southwestern Alpine beech forest conditions.

Materials and methods

Study area

The study was conducted in the southwestern European Alps across the neighboring regions of Canton Ticino (Switzerland) and Piedmont (Italy). The area is characterized by a marked elevation gradient ranging from Lake Maggiore (197 m a.s.l.) to Adula Peak in Ticino (3402 m a.s.l.) and to Punta Nordend in Piedmont (4609 m a.s.l.), respectively. The geology is characterized by the tectonics of the Alps with granite and gneiss dominating the bedrock (Pfiffner 2015). Due to the relief-rich terrain, rockfalls are one of the major natural hazards threatening mountain settlements and roads in both regions (Regione Autonoma Valle d'Aosta 2010; Ambrosi and Thüring 2005).

The regional climate can be described as warm and humid, with a high annual precipitation gradient ranging from 778 mm in Piedmont (climate station Susa: 07°3′0″E, 45°08′0″N; Arpa Piemonte 2015) to 1897 mm in Ticino (climate station Locarno Monti: 08°47′43″E, 46°10′12″N; MeteoSwiss 2015). More than half of the annual precipitation falls during the transition seasons between April–

May and September–November. In winter (December–March), precipitation is particularly low with 162 mm for Piedmont, and 316 mm for Ticino. Winters are generally mild with mean January temperatures around 3.5 °C, and summers are warm with mean July temperatures around 21.7 °C. Summers are humid with dry periods not lasting longer than thirty consecutive days (Isotta et al. 2014), whereas winters are dry, especially during North Foehn weather conditions. The relative humidity drops below 20 % in average on 40 days per year, when the Foehn wind travels south from the northern Alps (Spinedi and Isotta 2005). These North Foehn winds dry the fine flammable material of the forest understory and increase the risk of fire. Forest fires are mostly of human origin, consisting of surface fires in the understory of the deciduous forests. Those fires usually start from the urban–forest interface (Conedera et al. 2015) and rarely spread to the higher elevated beech belt (900–1500 m a.s.l.; Pezzatti et al. 2009).

Selection of burns and data collection

Fire perimeters <40 years old were selected from the forest fire databases of Switzerland (Pezzatti et al. 2010) and from the State Forestry Corps of Italy (Inventorio nazionale delle foreste e dei serbatoi di Carbonio (INFC 2005), Corpo Forestale dello stato—ispettorato generale). They were overlaid with local vegetation (Ceschi 2006; Camerano et al. 2004) and geological maps, using a geographical information system (ArcGIS version 10.0; © ESRI) to identify fires in beech stands on crystalline bedrock. The first field observation took place in 2011 to identify potential study sites that met all of the following criteria: (1) area burnt in beech forests >0.25 ha, (2) no signs of pre-fire pasture or post-fire artificial plantation, and (3) area dominated by beech (stem density >95 %) before the fire event. From the initial 94 potential sites, 34 satisfied all of the selection criteria and were retained for the field survey in the years 2012 and 2013 (“Appendix” in Table 3).

Depending on the area burned, we placed one to three transects, spaced 50 m apart in elevation, from the unburnt to the burnt beech forests (Fig. 1). Circular plots of 200 m^2 were placed regularly with 30 m distances in between starting at a distance of 10 m from the burn edge and following the contour lines. Whenever possible, a minimum of one control plot was placed in the unburnt beech forests at a distance of 20 m from the burn edge.

Data collection followed the guidelines of the Swiss National Forest Inventory (NFI; Keller 2005) with specific focus on the stand stability parameters (Herold and Ulmer 2001). Therefore, general plot characteristics were surveyed like slope (°); aspect; elevation (m a.s.l.); microrelief (plane, concave, convex); the cover of inhibitors for emerging regeneration such as common bracken (*Pteridium aquilinum* [L.] Kuhn), common broom (*Cytisus scoparius* [L.] Link), and purple moor grass (*Molinia arundinacea* Schank); as well as the surface roughness in the form of deposited rocks (see Brauner et al. 2005). The coverage of common bracken, common broom and purple moor grass was totaled per plot (hereafter referred to as “cover of early post-fire colonizers”).

We inventoried all trees with diameter to breast height (DBH) $\geq$8 cm and tree height $\geq$2 m. Smaller trees were omitted because of their negligible role in the protective effectiveness (Wehrli et al. 2006). Each standing tree was identified down to the species level (Wagner et al. 2010) and the following characteristics were recorded: vitality, i.e., tree being alive or dead (snags and dead standing tree with crown but without visible green foliage, hereafter referred to as snags), DBH (at 1.30 m to the nearest cm), tree height (to the nearest meter), and the percentage of crown volume killed. The latter was visually estimated by the volumetric proportion of crown killed compared to the space occupied by the pre-fire crown volume (Hood et al. 2007). Data collection further included fallen dead trees (hereafter referred to as logs) of which the average diameter and the length were recorded. For both snags and logs, the wood decay stage was recorded in four classes: (1) cambium still fresh, (2) knife penetrates low, cambium disappeared, (3) knife penetrates into the fiber direction, but not transversely, or (4) knife penetrates in both directions. Fallen branches and brushwood, originating from falling crowns of dead trees with a decay stage below four, were assessed using the method of Brown (1974). Pieces in the 200 m^2-plots were recorded in different diameter classes (1: 2.5–5 cm, 2: >5–7.5 cm, 3: >7.5–15 cm, 4: >15–30 cm) along the four cardinal directions. The resultant volume was then scaled up to standard hectare values (m^3 ha^{-1}).

In regions with such a relief-rich terrain fires burn very heterogeneously. Therefore, we categorized each plot according to low, moderate and high burn severity (burn severity refers to the long-term fire effects; cf. Morgan et al. 2014). In accordance to Maringer et al. (in press), we assessed burn severity by calculating the ratio of post-fire and pre-fire basal area of living trees. For burns older than 10 years, pre-fire conditions were assessed exclusively from the control plots, because of fast decaying dead wood in case of fire. On the other hand, in burnt plots younger than 10 years, the number of visible dead trees determined the pre-fire stand characteristics. Based on this assumption, we defined low burn severity in plots with <5 % crown volume loss and less than 20 % basal area loss. High burn severity was indicated by extensive crown loss (>50 %) and large basal area loss (>60 %). All plots with intermediate losses, in terms of crown and basal area, were

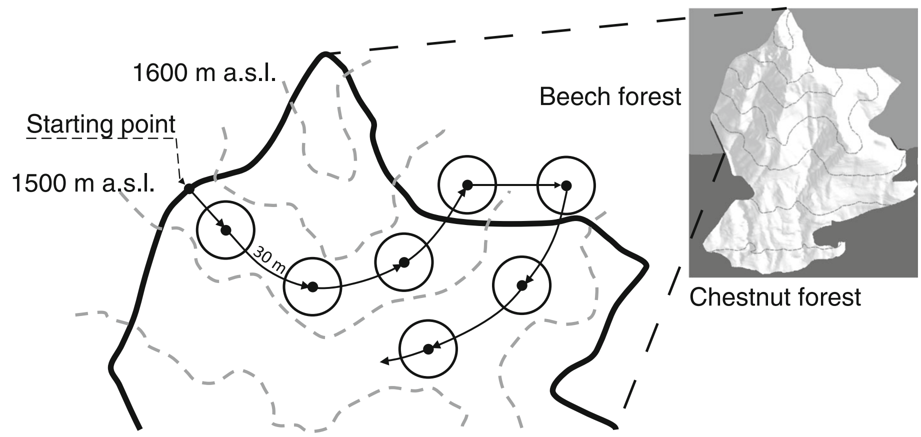

Fig. 1 Sampling design in burnt and unburnt beech forests with regularly placed circular 200 m^2 plots located 30 m apart along horizontal transects

assigned to the moderate severity class (for examples in the different severity burns see supplementary materials SM 1 to SM 3).

Analysis methods

The Rockfor.net model

Among available modeling approaches to assess the protective capacity of forests against rockfall, the energy line concept originally developed by Heim (1932) has proven to be a robust method to generally estimate rockfall runout zones and energy intensity (cf. Jaboyedoff and Labiouse 2011). Also based on the energy line concept, the rockfall hazard assessment model Rockfor.net additionally integrates the protective effect of forests (Berger and Dorren 2007). As a consequence, the tool quantifies the protective capacity of differently structured forest stands for various topographical settings as demonstrated in many applications in the European Alps (Berger and Dorren 2007; Wehrli et al. 2006; Kajdiž et al. 2015; Dorren et al. 2015). Ultimately, we chose to use this model because of its particular sensitivity to the dendrometric tree characteristics (such as fire-injured stems).

The underlying idea of the Rockfor.net model is to compare the theoretical basal area required for absorbing the kinetic energy of downhill moving rocks (G_{required}) and the available basal area of a particular forest stand ($G_{\text{available}}$). Therefore, the model regards all standing trees distributed in a forest as virtual tree lines parallel to the contour lines. All simulated trees have the same species composition and diameters (weighting of the tree species see Dorren and Berger 2005), representing the mean values in the original forest stands. The model starts by calculating the total kinetic energy developed by a rock falling down the slope. It then calculates the capacity of each tree line to dissipate energy. The number of trees required to dissipate all kinetic energy is subsequently converted in a required basal area (G_{required}) using the mean DBH. In the last step the Rockfor.net model quantifies the protective effect of a forest stand by comparing the required theoretical G_{required} with the available $G_{\text{available}}$ (see Berger and Dorren 2007 for more details).

In this study, we implemented the effect of fallen logs in the Rockfor.net by converting the related volume into a total log length per hectare and finally into the number of potential logs impacts per hectare. Here we assume that an efficient rock–log contact (defined as that with a rock/log diameter ratio of 1 or smaller) is required every 10 m on a slope length of 100 m to stop the falling rock completely. This approach is based on the small amount of data that exists on rockfall-log interactions, as collected during the experiments described in Dorren et al. (2015). The percentage of rocks stopped by logs ($\%R_{\text{stopped}}$) was calculated as follow:

$$\%R_{\text{stopped}} = \text{Eff}_{\text{contact}} \times \text{Vol}_{\text{Log}} \div \left(\pi \times \left(\frac{D_t}{2}\right)^2\right) \div 100\ \text{m} \div 10 \times 100\,\% \quad (1)$$

where, $\text{Eff}_{\text{contact}}$ = rock–log contact efficiency = min[1, D_t/D_b], D_t = mean log diameter in stand (in m), D_b = rock diameter (in m), Vol_{Log} = volume of fallen logs (in $m^3\ ha^{-1}$).

In sum, the Rockfor.net model requires as input parameters both site and forest stand characteristics. Required site characteristics are cliff height (m), length of both the forested and unforested slope on the trajectory of a fallen rock, and mean slopes inclination (°). Species composition, DBHs and densities of standing trees (including snags) as well as diameter and length of the logs (with a wood decomposition rate below four) are required as stand characteristics.

The contribution of lying branches and brushwood to rockfall energy dissipation is hard to quantify in a model such as the Rockfor.net and was therefore neglected for the purposes of this study. However, temporal changes in fallen branches and brushwood volume were graphically visualized (see "Surface unevenness" section).

Input data preparation and scenario specification

Data preparation followed the new rockfall protection guidelines of the 'Sustainability and success monitoring in the protection forests of Switzerland (NaiS)' (see Frehner et al. 2005 and Dorren et al. 2015). Tree diameters were grouped in four DBH-classes (8–12, 12–24, 24–36, and ≥36 cm) separately for living and dead standing trees and standardized to number of stems per hectare. Large-diameter trees most effectively dissipate the kinetic energy of falling rocks, especially those of large rocks, whereas small-diameter trees significantly increase the probability of rock–tree contacts due to (generally) large stem densities. Therefore, the required basal area (G_{required}) to stop a falling rock within a specific forested slope is weighted for the DBH-classes according to the rock size (Dorren et al. 2015). Moreover, to account for the differences in capacity of different tree types to dissipate the kinetic energy of falling rocks, Rockfor.net converts the proportions of the presence of five different tree 'types' in each stand into a mean dissipative energy capacity per study site. The following tree 'types' were taken into account: beech, Norway spruce (*Picea abies* [L.] Karst.), and silver fir (*Abies alba* Mill.). The remaining broadleaved and conifer trees were merged in two separate groups (cf. Dorren and Berger 2005).

Further, our research used standardized rock sizes and mean slope gradients, cliff heights and lengths of the forested slopes. We defined standard rock volumes (0.05, 0.2, and 1 m^3, which corresponds to the rock diameters 0.37, 0.58 and 1 m; Table 1) as traditionally used in NaiS (Frehner et al. 2005; Dorren et al. 2015). In order to simulate realistic field conditions, we defined two options of horizontal distances (75, 150 m)—from the bottom of a cliff to the downslope forest edge—in which a rock had to be stopped. Finally, three different slope gradients were considered representing the 1st (27°) and 3rd (35°) quantiles, and mean (30°) of the slope distribution from the surveyed plots (Table 1). Slope gradient was standardized after testing the statistical non-significance between tree stem densities and slopes using a mixed-effect model (Table 4 in Appendix).

Table 1 Scenario specifications for the Rockfor.net model

Input parameters	Scenario specifications					
Cliff height (m)	20					
NFS (m)[a]	0					
Rock density (kg m^{-3})	2800					
Forested slope length (m)	75			150		
Slope inclination (°)	27	30	35	27	30	35
Rock volume (m^3)	0.05	0.05	0.05	0.05	0.05	0.05
	0.2	0.2	0.2	0.2	0.2	0.2
	1	1	1	1	1	1

[a] *NFS* non forested slope length between the foot of the cliff and the upper forest limit

The estimation of the protective effect as calculated by the Rockfor.net model represents the probability of a rock to be stopped in a beech stand. This is expressed in the following categories: ≥90 % very good protection, 75–90 % good protection, 50–75 % adequate protection, 25–50 % moderate protection, and <25 % inadequate protection. Whether or not the level of protection provided by a forest stand is sufficient, it can only be determined by means of a risk analysis in which the effective risk reduction of the forest is quantified; this level of analysis is therefore out of the scope of the present paper.

Analysis of the modeled results

The protective capacity for each scenario was given as the proportion of rocks stopped by standing trees (living and dead) and by logs at the plot level. The result was set to 100 % in case the sum exceeded this mark. The temporal trends in the protective capacity in mixed-severity burns for different scenarios were visualized using standard loess-smoothing curves (Chambers and Hastie 1992). The corresponding unburnt beech forests served as reference. The loess-smoothing curve is a flexible tool making it easy to model complex processes, but it does not allow for one easily interpretable regression function. Therefore, we additionally employed linear regression models with protective capacity as the response variable and the number of post-fire years as the explanatory variable. Since the protective capacity is expressed as percentage (probability), the data were log-transformed $\left(y' = \log\left(\frac{y}{1-y}\right)\right)$ and the numbers of post-fire years were included as linear and quadratic terms. Additionally, Mann–Whitney–Wilcox tests were applied in each of the calculated scenarios for detecting significant differences in distributions of the forest protective capacity in different severity burns and the corresponding unburnt beech forests.

All analyses of the modeled results and the regression models were performed using *R*, the free software environment for statistical computing (R Development Core Team 2014). Negative binomial logistic regression models were fitted and validated using the glmmADMB package (Bolker et al. 2012). Graphical outputs are mainly based on the packages lattice (Deepayan 2008) and ggplot2 (Wickham and Chang 2015).

Results

Forest characteristics and development after fire

We assessed a total number of 189 plots in burnt and 27 plots in unburnt (control plots) beech forests. Most of the burnt plots were classified as moderate- (44.2 %) and high-severity burns (40.3 %), whereas only the remaining 15.5 % were considered as low-severity burns. Elevation of the burns and the corresponding unburnt beech forests ranged from 700 to 1486 m a.s.l. with mean slope inclinations of 30 ± 0.34°.

Beech grew frequently in the burnt forests, with percentages ranging from 21 to 100 % ("Appendix" in Table 3). Other broadleaved species were only present in two-thirds of the investigated burns, and their frequency was mostly lower than that of beech. Rarely conifer tree species grew in the burns and their proportion never exceeded 21 %. The overall average tree height was 10 ± 0.1 m, and approximately 2 m higher when referring to living trees only.

Average densities of both living and dead standing trees was 716 ± 27 stems ha^{-1} in the burnt beech forests, and 883 ± 48 stems ha^{-1} in the unburnt beech forests. The average proportion of snags varied according to the burn severity from 19 % in low-severity burns, to over 40 % in moderate-severity burns, and up to 61 % in high-severity burns (Fig. 2). Correspondingly, the volume of logs increased from 10 ± 3 m^3 ha^{-1} in low-severity burns to 52 ± 9 m^3 ha^{-1} in high-severity burns.

In the examination of the impact of mixed-severity fires on trees, it was revealed that small-diameter trees mostly suffer in low-severity burns (23–53 %), whereas large-diameter trees survive. Log volumes were below 10 m^3 ha^{-1}, except 16–20 years post-fire (Fig. 2).

The proportion of dead standing trees increased in moderate-severity burns, and not only for small- and intermediate-diameter trees but also for large-diameter trees (Fig. 2). Most trees died within the first 20 years post-fire, leaving space for new regeneration—as indicated by the increasing number of small-diameter trees from 20 years post-fire onwards. The volume of logs was up to six times higher (27 and 50 m^3 ha^{-1}) in early post-fire stages (≤20 years post-fire) than for those recorded in later periods (>20 years post-fire).

Tree mortality was high throughout all diameter classes in high-severity burns, mostly within the first 20 years post-fire (Fig. 2). During this period, the volume of logs peaked with 62 m^3 ha^{-1}. From 20 years post-fire onwards, the volume of logs and the proportion of dead standing trees halved and new regeneration rapidly increased doubling in density as compared to moderate-severity burns.

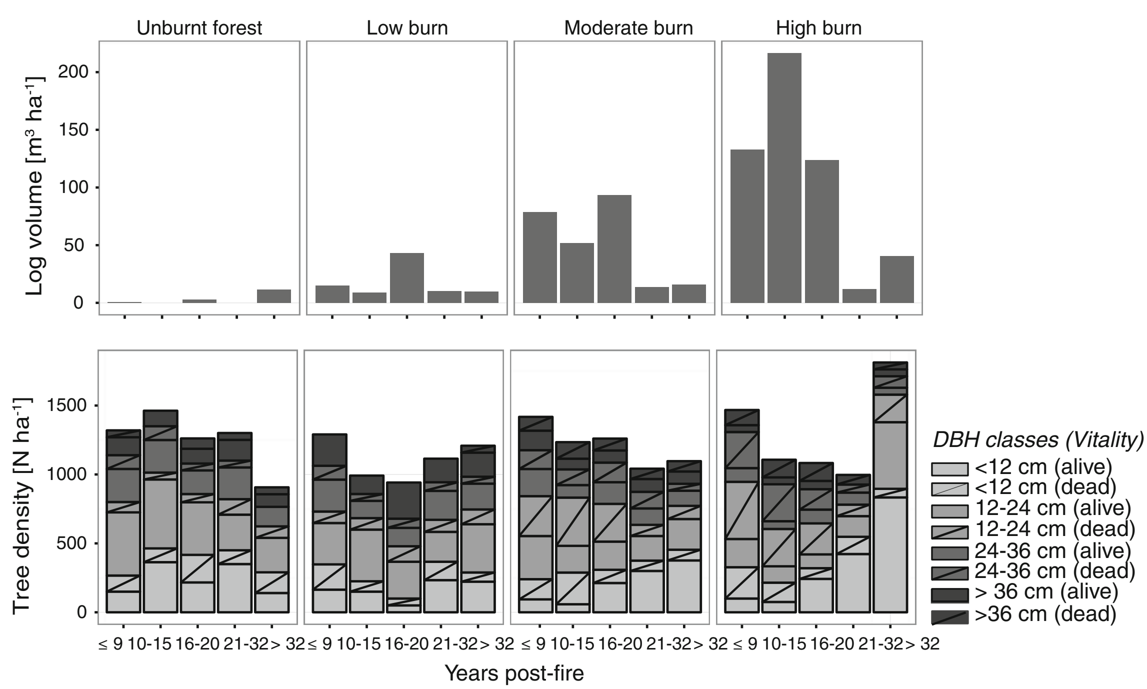

Fig. 2 Volume of logs and tree densities (DBH ≥8 cm) for living and dead (*shaded bars*) trees in different DBH-classes (*gray color* gradient) for low-, moderate- and high-severity burns and the corresponding unburnt beech forests, grouped by years post-fire

Surface unevenness

Most burnt plots (46 %) were located on a plane surface, whereas 31 % were in small depressions, and the remaining 23 % on convex micro-relief. The average coverage of rocks in a burnt plot was 2 %, ranging from 0 % to a maximum of 30 %. Early post-fire colonizers grew prolifically, reaching an average coverage of 28 % in moderate-severity burns and 56 % in high-severity burns (Fig. 3). Maximum coverage (∼60 %) was reached by around 30 years post-fire in high-severity burns, which was more than double that in moderate-severity burns. Coverage of early post-fire colonizers hardly exceeded 25 % in low-severity burns and was close to zero in the unburnt beech forests.

Patterns in the volume of fallen dead branches and brushwood were similar across different severity burns with peaks at around 15 years post-fire (Fig. 4). Volumes steadily decreased over time, and reached nearly normal volumes (similar to control plots) after 30 years post-fire.

The volume of fallen branches and brushwood scored the highest average values (106 m^3) in high-severity burns, where it was 1.5-times higher than in moderate- (75 m^3) and low-severity (60 m^3) burns. Contrastingly, no clear temporal trend was detected in the unburnt beech forests where volumes of fallen branches and brushwood never exceeded 25 m^3 ha^{-1}.

Temporal trends in the protective capacity of forests

The Rockfor.net model results highlight the mid-term (first 40 years post-fire) evolution of the protective capacity of burnt beech stands as a function of different burn severities, rock sizes, forested slope lengths, and slope inclinations. The average protective capacity aggregated over the years post-fire decreased when the rock size and slope inclination increased, and the length of the forested slope decreased (Table 2). The protective capacity of low-severity burns did not significantly differ from the unburnt beech forests for most of the scenarios. However, for moderate- and high-severity burns, the protective capacity significantly differed from the unburnt beech forests in more than half (67 %) of the scenarios (Table 2).

Low- and moderate-severity burns yielded a protective capacity above 50 % (more than adequate) for small and intermediate-sized rocks regardless of the forested slope length (Figs. 5, 6). Only in scenarios with rocks of 0.2 m^3, slope inclination $\geq 30°$ and forested slopes length shorter than 75 m did the protective capacity decrease below 50 %, mostly between 20 and 30 years post-fire (Fig. 6a). In similar scenarios, the protective capacity in high-severity burns ranged between ∼10 % (inadequate) and 45 %, and was at a minimum in scenarios combining intermediate-sized rocks with steep and short forested slopes (Fig. 6a).

For scenarios with rocks of 1 m^3 and 150 m forested slopes, the protective capacity of the forests was above 50 % (adequate protection) for unburnt beech forests and for low-severity burnt beech forests without any clear temporal trend (Fig. 7b). In the case of shorter forested slopes, the protective capacity of those forest types ranged between 25 % (satisfying) and 75 % (adequate) (Fig. 7a). Contrastingly, the protective capacity in moderate- and high-severity burns younger than 15 years post-fire rapidly decreased below 50 %, reaching its minimum (∼10 % that is inadequate) around 20 years post-fire. As a general rule,

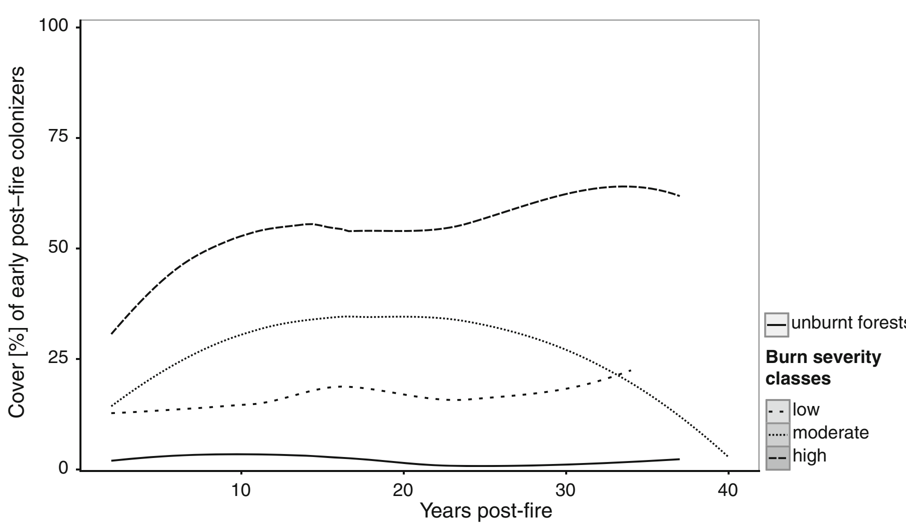

Fig. 3 Temporal trends for the cover of early post-fire colonizers (sum of *Pteridium aquilinum, Cytisus scoparius, Molinia arundinacea*) visualized by loess-smoothing curves (*black dotted lines*) for the different burn severity classes and the corresponding unburnt beech forests

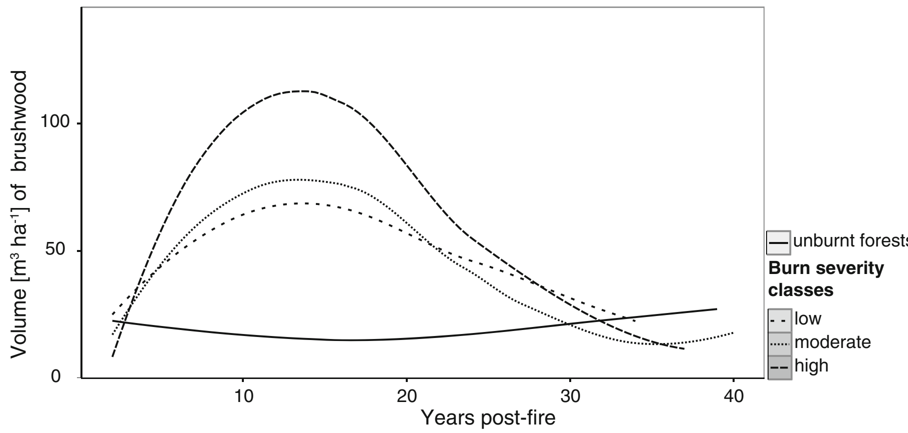

Fig. 4 Temporal trends in the volumes ($m^3\ ha^{-1}$) of fallen dead branches and brushwood, visualized by loess-smoothing curves for the different burn severity classes and the corresponding unburnt beech forests

Table 2 Mean protection capacity (%) for the different scenario specifications grouped by low, moderate and high burn severity and the corresponding unburnt beech forests

Forested slope length		75 m			150 m		
Mean slope		27°	30°	35°	27°	30°	35°
Rock size	Burn severity	Mean protective capacity (%)					
$0.05\ m^3$	Unburnt	97	95	91	95	95	95
	Low	96ns	92ns	87ns	92ns	92ns	92ns
	Moderate	89ns	85*	76**	88*	87*	87*
	High	73*	68**	61**	74*	73**	69**
$0.2\ m^3$	Unburnt	94	84	69	95	94	89
	Low	87*	83ns	71ns	94ns	91ns	84ns
	Moderate	77**	66ns	57*	89ns	85*	71*
	High	55***	49**	40***	73*	67**	53*
$1\ m^3$	Unburnt	62	48	30	94	75	58
	Low	61ns	54ns	37ns	93ns	76ns	56ns
	Moderate	47**	37ns	28ns	86ns	59*	39*
	High	33***	28**	23ns	65**	41***	29**

Similarities (Mann–Whitney–Wilcox tests) in the protection capacity between unburnt and burnt beech forests of different severities are shown in the superscript

Signif. codes: '***' 0.001; '**' 0.01; '*' 0.05; '.' 0.1 'ns' 1

the log components contributed only very minimally to the overall protective effect of the analyzed scenarios (see supplementary materials SM 22 and SM 23). (For details on single scenarios and single plots data, refer to supplementary materials SM 4 to SM 21.)

The linear regression models applied to detect temporal trends in the protective capacity of the burnt and unburnt beech forests showed significant correlations between the protective effect and the linear and quadratic terms of the number of post-fire years for most of the moderate- and high-severity burn scenarios. Such a significant correlation was missing for low-severity burns and the unburnt beech forests (Table 5 in "Appendix").

Discussion

Pattern in the protective capacity

The protective effect of forest stands against rockfall highly depends on species composition, stand structure, and forest regeneration capacity (Motta and Haudemand 2000; Dorren et al. 2004a; Dorren and Berger 2005). Disturbances such as forest fires abruptly and substantially change the forest structure, which may temporarily affect the protective capacity of the concerned forest stand (e.g., Dorren et al. 2004a; Vacchiano et al. 2016).

Our results show that in beech-dominated stands, episodic surface fires cause little changes in tree species composition. Beech directly re-grows (Maringer et al. in press) after single fire events, resulting in stable and locally adapted forest on the long term (Dorren et al. 2004a; Rigling and Schaffer 2015).

However, the post-fire vertical and horizontal stand structures, as well as the amount and timing of regeneration, strongly depend on the burn severity. The forest structure in low-severity burns is mostly comparable to those of the unburnt forests (Keyser et al. 2008). Small, fire-related changes in tree density, canopy layer, and regeneration dynamics do not seem to affect the overall protective effect. This contrasts to moderate- and high-severity burns, where significant structural changes occur after fire. This may cause temporary failures in the protective capacity against rockfall, depending on the forested slope length, the mean slope gradient, and the rock size. Structural changes in moderate-severity burns are mostly due to the dieback of small- and intermediate-sized beech trees, which goes in line with post-fire observations in conifer stands (Keyser et al. 2008). Surviving large-diameter trees maintain protective capacity to some extent (Volkwein et al. 2011) and provide seeds for regeneration at the same time. The gradual opening of the tree canopy leads to emerging beech regeneration (Maringer et al. in

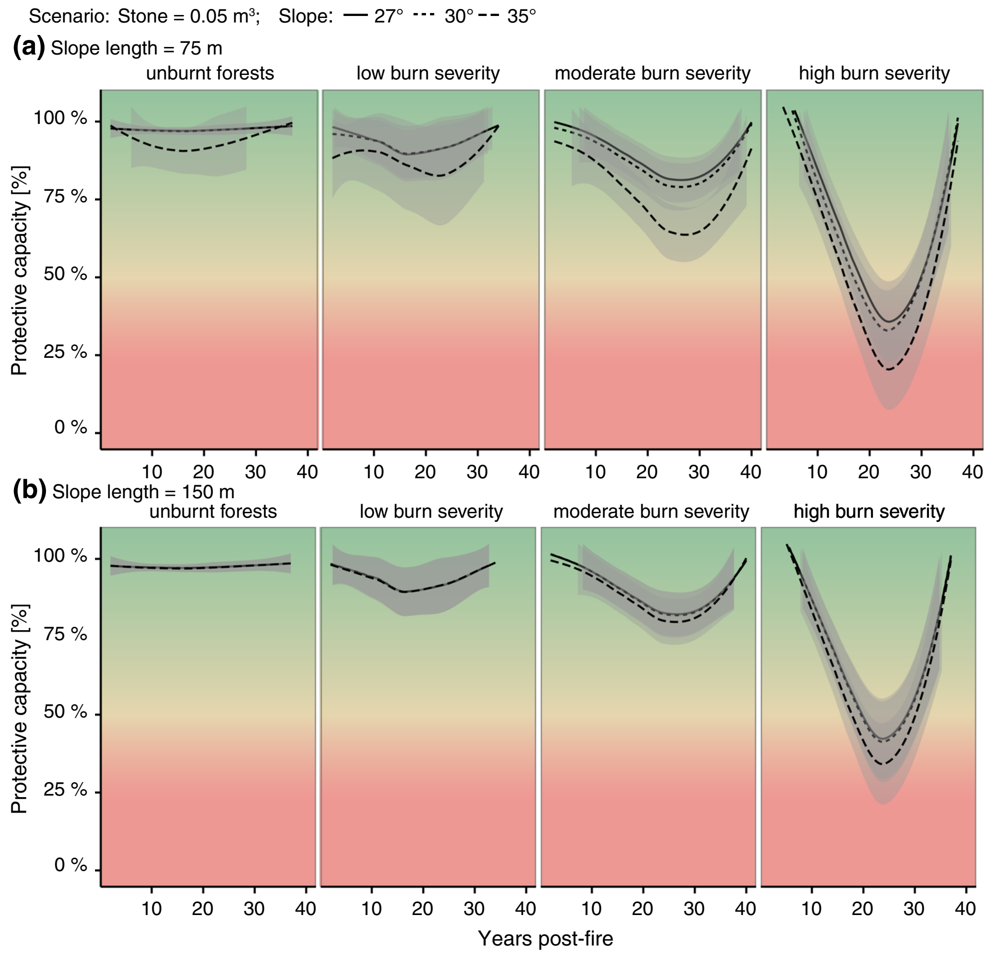

Fig. 5 Temporal trends in the protective effect (%) of beech stands in low-, moderate- and high-severity burns and the corresponding unburnt beech forests against small rocks (0.05 m^3), 75 m (**a**) and 150 m (**b**) forested slopes

press), so that the forest protective effect increases again after 20 years post-fire. In the long-term, the mixture of surviving tall and emerging small- and intermediate-sized beech trees results in a multi-layer stand structure that may better meet the protective function standards than mono-layered stands (Dorren et al. 2005b; O'Hara 2006). Nevertheless, the temporary deficit in the protective effectiveness of the burnt beech forests seems to occur between 10 and 35 years post-fire, especially in the case of forested slopes limited in length.

The majority of beech mortality in high-severity burns occurs within the first 20 years post-fire and concerns all tree sizes. This is similar to crown fires in conifer stands (Keyser et al. 2008; Brown et al. 2013) and to windthrow areas, where most trees die immediately after the disturbance event. In those areas, standing and fallen dead trees mostly maintain a protective effect (Frey and Thee 2002; Schönenberger et al. 2005; Bebi et al. 2015), although their resistance decreases with time since disturbance, as shown by tensile tests (Frey and Thee 2002; Bebi et al. 2012, 2015). The dead wood quantity and quality might also be lower in burns than in windthrow areas (Wohlgemuth et al. 2010; Priewasser et al. 2013), especially in the case of rapidly decaying tree species like beech (Kahl 2008). As shown by our results, the amount of dead wood consistently decreases from 15 years post-fire onwards, contributing little in the long-term to the forest protective capacity (Frey and Thee 2002). Such a loss in protective capacity has to be compensated by regeneration, which might be delayed due to a lack of seed providing trees and/or an exuberant layer of competing, fast-growing early post-fire colonizers. The latter are able to prevent immediate post-fire beech regeneration (Herranz et al. 1996; Ascoli et al. 2013; Maringer et al. in press), inhibiting forest re-growth for several decades (Koop and Hilgen 1987). At the same time, our results indicate a significant increase in the coverage of early post-fire colonizers and fallen dead branches, which may to some extent contribute to the protective capacity against falling rocks with volumes smaller than 0.2 m^3 in the first 20 years post-fire.

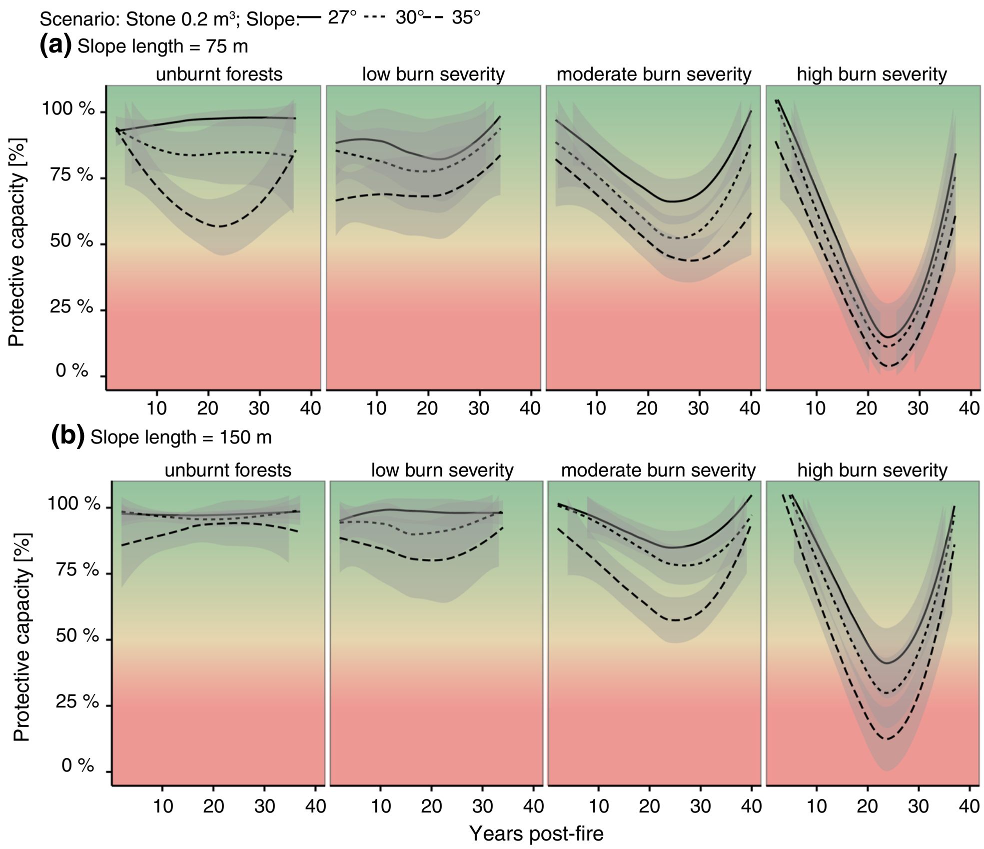

Fig. 6 Temporal trends in the protective effect (%) of beech stands in low-, moderate- and high-severity burns and the corresponding unburnt beech forests against intermediate-sized rocks (0.2 m^3), 75 m (**a**) and 150 m (**b**) forested slopes

However, to date their effective contribution is hard to quantify even in process-orientated models.

Limits of the study

The initial motivation of the present study was to test the general protective capacity of beech forests exposed to fires of varying severity. We asked whether a general drop in the protective capacity results from structural changes due to the dieback of fire-injured beech trees and to a delay in beech regeneration. We choose to use the Rockfor.net model because of its particular sensitivity to structural forest parameters.

We calculated the protective capacity of burnt beech stands for standard conditions and thus independently from local plot conditions (e.g., topography, soil type). In doing so, we provided foresters with a tool to quickly estimate decreases in protection capacity based on the burn severity and average characteristics of the concerned site. A precise assessment of the probability of stopping falling rocks in a particular forest stand, however, has to be conducted by implementing detailed local site conditions (e.g., slope inclination, length of the forested slope) or by using a spatially explicit model with high resolution (e.g., Zinggeler et al. 1991; Dorren 2016; Bartelt et al. 2002).

Furthermore, we were not able in this study to include in rockfall modeling the contribution of brushwood and early post-fire colonizers. Since they might contribute to protective capacity in disturbed forests more than in undisturbed forests, further research is needed in order to quantify their potential to dissipate energy. Such research would help to better align model-based assumptions with the effective situation in disturbed forests.

Conclusion and practical consequences for forest managers

In this paper we analyze temporal trends in the capacity of forests to protect against rockfall, focusing on burnt beech stands in the southwestern Alps. Based on our results, standing or fallen dead trees should in general be left at the burn site not only because they provide seeds, shade, moisture and nutrients to the emerging tree regeneration (Maringer et al. in press) but also because of their temporal contribution to the forest's capacity to protect against

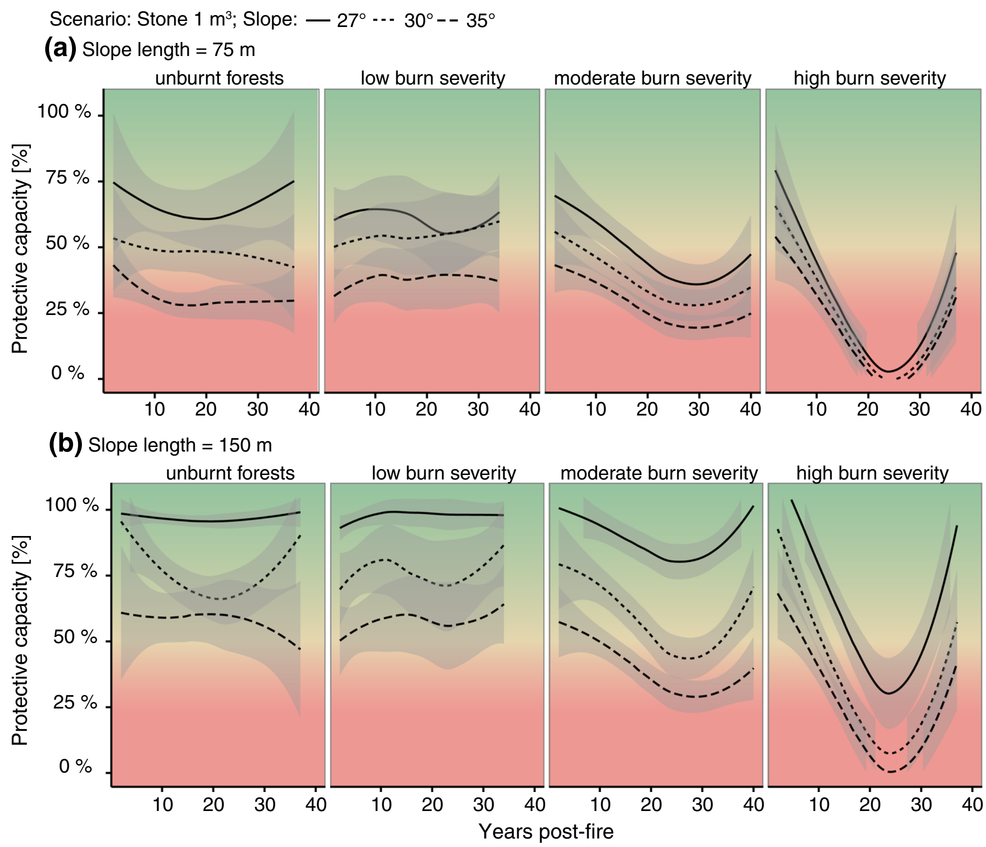

Fig. 7 Temporal trends in the protective effect (%) of beech stands in low-, moderate- and high-severity burns and the corresponding unburnt beech forests against large rocks (1 m^3), 75 m (**a**) and 150 m (**b**) forested slopes

rockfall. In particular, burnt beech forests hit by low-severity fires provide nearly the same protective effects as unburnt ones. Hence, silvicultural measures are generally not necessary, whereby the protective capacity has to be assessed on an individual basis.

In the case of moderate-to-high severe fires, stands may experience a temporary deficit in their protective capacity between 10 and 30 years post-fire, depending on the effective burn severity, the rock sizes, the length and the mean inclination of the forested slope. The cumulative effect of both the dieback of fire-injured trees and the delayed and slow re-growth of the regeneration may reduce protective capacity to below 50 %, especially in the case of large falling rocks on steep slopes. Consequently, silvicultural and/or technical measures may be necessary in such critical scenarios depending on the risk for humans and their property in relation to the cost–benefit ratio. Besides the installation of rockfall nets or walls, small-scale directional felling of standing dying trees and oblique positioning of the resulting logs offers a possibility to temporarily mitigate the loss of protective capacity. However, the felling has to be conducted within a particular timeframe, because (1) the time-lag between salvage logging and a beech mast year affects the regeneration process (Ascoli et al. 2015), and (2) beech wood decays relatively quickly over time (Ascoli et al. 2013) and in the case of secondary fungi infestation (Maringer et al. 2016), which enhances the mechanical instability of trees and makes increasingly difficult the implementation of directional felling. As mentioned by Ascoli et al. (2013, 2015), salvage logging should be carried out in the winter following a beech mast year, because the success of beech regeneration is highly dependent on quantitative seed input. Interventions later than 5 years post-fire should be avoided to protect established beech saplings. Moreover, weed control combined with artificial beech seed dispersal could reduce the inter-species competition and may accelerate the establishment of a new beech generation.

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Appendix

See Tables 3, 4 and 5.

Table 3 Investigated burns sorted by the date of fire

Location	Site characteristics				Burnt forests characteristics				Control
Municipal	Date of fire	Slope	Elev.	Nr. plots	Mean stem density	Mean basal area	Species	Species proportion	Nr. plots
Gordevio	09.03.73	24	1460	1	1900	10	F.s.	97	0
							Broad.	3	
Moghegno	27.11.73	40	1100	3	883	38	F.s.	50	0
							Broad.	50	
Arbedo	20.03.76	31	1300	13	912	36	F.s.	76	1
							Broad.	22	
							P.a.	3	
Godevio	28.03.76	6	1060	2	350	37	F.s.	100	1
Sparone	28.12.80	22	1100	16	753	27	F.s.	62	1
							Broad.	37	
							Conif.	1	
Astano	01.01.81	22	1050	2	750	35	F.s.	70	0
							Broad.	30	
Indemini	01.01.81	31	1200	12	613	13	F.s.	71	1
							Broad.	29	
Intragna	04.01.87	27	1150	3	583	18	F.s.	100	0
Aurigeno	01.08.89	35	900	2	1500	25	F.s.	84	1
							Broad.	16	
Corio	15.02.90	19	1080	10	295	26	F.s.	60	2
							Broad.	40	
Mugena	23.03.90	19	900	6	108	29	F.s.	100	1
Novaggio	10.03.90	35	1300	2	225	8	Broad.	38	1
							F.s.	62	
Rosazza	19.01.90	40	1000	5	460	49	F.s.	91	0
							Broad.	9	
Avegno	05.05.90	20	1250	2	50	19	F.s.	75	0
							Broad.	25	
Pollegio	09.04.95	22	1200	3	117	22	F.s.	56	2
							Broad.	44	
Tenero	21.04.96	37	950	3	200	15	Broad.	18	0
							F.s.	82	
Arola	04.06.97	40	800	13	646	37	F.s.	66	0
							Broad.	34	
Magadino	15.04.97	33	1200	24	427	28	F.s.	72	3
							Conif.	2	
							Broad.	26	
Ronco s. A.	15.03.97	22	1300	6	417	23	F.s.	100	1
Sonvico	03.04.97	24	1000	5	380	13	F.s.	49	2
							Broad.	51	
Arbedo	14.11.98	33	1350	3	250	10	F.s.	100	2
Indimini	19.12.98	33	1300	1	100	30	F.s.	50	1
Gordevio	24.04.02	24	1400	5	490	31	F.s.	100	4
Maggia	11.03.98	14	1380	3	617	32	F.s.	100	1
Bodio	17.03.99	33	1050	3	167	48	F.s.	62	1
							Broad.	38	
Dissimo	05.04.99	40	1000	3	900	27	F.s.	97	1
							Broad.	3	

Table 3 continued

Location	Site characteristics				Burnt forests characteristics				Control
Municipal	Date of fire	Slope	Elev.	Nr. plots	Mean stem density	Mean basal area	Species	Species proportion	Nr. plots
Someo	05.08.99	27	1450	3	433	35	F.s.	100	1
Villadossola	15.03.01	37	1250	11	1009	27	F.s.	79	1
							Broad.	21	
Cugnasco	02.04.02	22	700	4	575	21	Broad.	53	1
							F.s.	47	
Ronco s.A.	22.04.03	3	1300	2	350	35	F.s.	100	1
Varallo	11.08.03	29	1300	11	323	26	F.s.	96	1
							Broad.	4	
Condove	01.03.08	19	1100	11	573	50	F.s.	98	1
							Broad.	2	
Drugno	26.03.12	29	1100	12	963	20	F.s.	90	1
							Broad.	10	
Giaglione	03.03.12	39	1300	8	994	44	F.s.	77	1
							Conif.	21	

Further listed: slope (°), elevation [elev. (m a.sl.)], number of plots, mean stem density [stems ha^{-1}], mean basal area (m^2 ha^{-1}), species (F.s. = *Fagus sylvatica*, Broad. = other broadleaf species, P.a. = *Picea abies*, Conif. = other conifer species, species proportion of living trees (%), number of plots in the corresponding unburnt forest (control)

Table 4 Estimates and standard error of the mixed-effect model for stem densities modeled against slope inclination

Variable	Estimate	Standard error
Intercept	5.9	<0.0001
Slope	0.009	0.25
Random intercept	Variance	Std. Dev.
	0.33	0.6

Slopes of the plots were measured in degree and implemented as the explanatory variable in a mixed-effect model with negative binomial distribution (Bolker et al. 2012). Stem densities served as the response variable, and because of the high intra-class correlation, burns were implemented as random effect in the model. The result shows that slope inclination was not significant at the 0.05-level (Table 4), and thus it was possible to use standardized slope inclination in the Rockfor.net tool. Against this background, the 1st (26.7°) and 3rd (35°) tertiles, as well as the mean (29.7°), were used as standardized slope inclinations.

Table 5 Linear regression models for temporal trends in the years post-fire (AGE) of the protective capacity of burnt beech stands differing in burn severity (low, moderate, high) and the corresponding unburnt forests

Scenario						
Rock size (m^3)	Forested slope length (m)	Slope inclination (°)	Burn severity	Intercept	AGE	AGE^2
0.05	75	27	Unburnt	(+)***	ns	ns
			Low	(+)**	ns	ns
			Moderate	(+)·	ns	ns
			High	(+)***	(−)**	(+)**
0.05	150	27	Unburnt	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)·	(+)·
			High	(+)***	(−)***	(+)***
0.05	75	30	Unburnt	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)·	(+)·
			High	(+)***	(−)***	(+)***

Table 5 continued

Scenario						
0.05	150	30	Unburnt	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)·	(+)·
			High	(+)***	(−)***	(+)***
0.05	75	35	Unburnt	(+)**	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)*	(+)·
			High	(+)***	(−)***	(+)***
0.05	150	35	Unburnt	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)·	(+)·
			High	(+)***	(−)***	(+)***
0.2	75	27	Unburnt	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)**	(+)**
			High	(+)***	(−)***	(+)***
0.2	150	27	Unburnt	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)*	(+)*
			High	(+)***	(−)***	(+)**
0.2	75	30	Unburnt	(+)**	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)***	(+)***
			High	(+)***	(−)***	(+)***
0.2	150	30	Unburnt	(+)***	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)*	(+)*
			High	(+)***	(−)***	(+)***
0.2	75	35	Unburnt	(+)***	(−)**	(+)**
			Low	(+)*	ns	ns
			Moderate	(+)***	(−)**	(+)*
			High	(+)***	(−)***	(+)***
0.2	150	35	Unburnt	(+)**	ns	ns
			Low	(+)***	ns	ns
			Moderate	(+)***	(−)***	(+)**
			High	(+)***	(−)***	(+)***
1	75	27	Unburnt	ns	ns	ns
			Low	ns	ns	ns
			Moderate	(+)**	(−)*	(+)·
			High	(+)***	(−)***	(+)***
1	150	27	Unburnt	(+)***	ns	(−)*
			Low	(+)***	·	ns
			Moderate	(+)***	(−)*	(+)*
			High	(+)***	(−)***	(+)***
1	75	30	Unburnt	ns	ns	ns
			Low	ns	ns	ns
			Moderate	(+)*	(−)**	·
			High	(+)***	(−)***	(+)***

Table 5 continued

Scenario						
1	150	30	Unburnt	(+)***	(−)**	(+)**
			Low	(+)*	ns	ns
			Moderate	(+)***	(−)**	(+)**
			High	(+)***	(−)***	(+)***
1	75	35	Unburnt	ns	(−)·	ns
			Low	(+)**	ns	ns
			Moderate	(+)*	(−)**	(+)·
			High	(+)**	(−)***	(+)***
1	150	35	Unburnt	ns	ns	ns
			Low	ns	ns	ns
			Moderate	(+)*	(−)**	(+)·
			High	(+)***	(−)***	(+)***

Models were separately conducted for scenarios differing in rocks size (0.05, 0.2, 1 m^3), forested slope length (75, 150 m) and slope inclination (27°, 30°, 35°). The sign and significance level of the predictor are displayed

Signif. codes: '***' 0.001; '**' 0.01; '*' 0.05; '·' 0.1; 'ns' 1

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